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Modelling the marine eutrophication: A review

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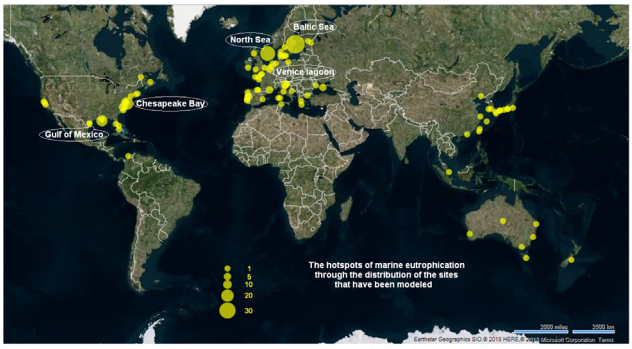
HIGHLIGHTS

- 291 references on marine eutrophication modelling have been analysed.
- 4 batches have been done: estuaries, lagoons, coastal seas, "green tides".
- The world hotspots of eutrophication are listed, with their main results.
- The evolution of tools, their strengths and weaknesses are described.

GRAPHICAL ABSTRACT



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ABSTRACT

In the frame of a national, joint scientific appraisal, 45 scientific French-speaking experts have been mandated in 2015–2016 by the French ministries of Environment and Agriculture to perform a global review of scientific literature dealing with the eutrophication phenomenon, in freshwater as well as in marine waters. This paper summarizes the main results of this review restricted to a sub-domain, the modelling approach of the marine eutrophication. After recalling the different aims pursued, an overview is given on the historical time course of this modelling effort, its world distribution and the various tools used. Then, the main results obtained are examined, highlighting the specific strengths and weaknesses of the present models. Needs for future improvement are then listed.

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1. Introduction

At the end of the 20th century, the Oslo and Paris Commission (OSPAR) in charge of promoting the good ecological status of West-European coastal seas proposed an operational definition of marine eutrophication (OSPAR, 1997). Eutrophication was then considered as "the result of excessive enrichment of water with nutrients which may cause an increase in the accelerated growth of algae in the water column and higher forms of plants living on the bottom of the sea." The OSPAR definition pointed out that eutrophication "may result in a range of undesirable disturbances in the marine ecosystem, including a shift in the composition of the flora and fauna which affects habitats and biodiversity, and the depletion of oxygen, causing death of fish and other species." The most mediated aspects of the marine eutrophication are the mass accumulation of green macroalgae on beaches (the so-called "green tides") or in lagoons, as well as the intense proliferations of some phytoplanktonic species in coastal seas (the so-called "colored waters"). However, massive kills of fishes and benthic fauna have also revealed the deleterious effects of the invisible anoxia of

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bottom waters, leading to sporadic or permanent "dead zones" in more and more places (Diaz & Rosenberg, 2008). Eutrophication can be a natural process: in inland waters, ageing of a lake over geological time scales induces a slow accumulation of organic material, whereas in some oceanic areas, as off Namibia (Brüchert et al., 2003), intermittent and sudden massive inputs of deep, nutrient rich water upwelled by wind episodes can trigger a high surface production of phytoplankton, which sinks and feeds bottom anaerobic sulphate reducing bacteria. To embrace accumulation of allochtonous organic matter as well as sedimentation of locally produced algal material, (Nixon, 2009) defined eutrophication as an "increase in supply of organic matter". Most of present cases of marine eutrophication are from anthropogenic origin, i.e. caused by recent abnormal man-made inputs of nutrients or organic wastes. The study of many lakes, first affected by anthropogenic eutrophication in the second half of the 20th century, has shown (Vollenweider, 1968) that eutrophication is possible only if the surface mixed layer is sufficiently thin and illuminated to allow a primary production greater than the algal respiration and hence, to ensure a rapid algal growth. The biomass produced can accumulate only if the water body exhibits a residence time of several days, i.e. is sufficiently confined. Eutrophication in lakes revealed to have been mainly triggered by massive anthropogenic inputs of inorganic phosphorus (phosphate). In coastal seas however, ecosystems are naturally open, they show strong continuous gradients extending from the estuaries to the offshore waters. Their high salinities are not in favor of an efficient gaseous nitrogen fixation by cyanobacteria, so that the controlling role of nitrogen may override the phosphorus one. A part of the terrestrial loadings of nutrients is exported into the oceanic waters, gyres of various sizes embedded in the mainstream tidal residual flow provide spatially heterogeneous residence times, whereas oceanic oscillations, such as the North Atlantic Oscillation, can modulate the marine response to nutrient enrichment… This further adds complexity to the marine eutrophication process.

Scientists willing to understand the mechanism of eutrophication as well as public administrations aiming at defining the optimal actions for remediation have used mathematical models to explore hypotheses or remediation scenarios during the four last decades. The need for integrated actions from the watersheds down to the open ocean under a changing climate will probably increase the role of numerical models in tackling the eutrophication problem in the future. What help did the models provide up to now, what cannot they do at this moment, what would we like they should be able to do in the future? This paper tries to answer these questions from the existing literature.

2. Methods used to build the inventory

This work has been conducted in 2015–2016, as part of a global review of scientific literature dealing with eutrophication, in fresh inland waters as well as in marine coastal waters. The French Ministry of Ecological and Social Transition (MTES), together with the French Agency for Biodiversity (AFB) and the French Ministry of Agriculture and Food (MAA) mandated the French National Centre for Scientific Research (CNRS), along with three other French public scientific institutes (National Institute for Agronomic Research-INRA, French Institute for Research and Exploitation of the Sea-IFREMER, and Institute for Scientific Research and Technology for Environment and Agriculture-IRSTEA), to coordinate a panel of 45 scientific French-speaking experts which could cover all the different aspects of the eutrophication problem (biogeochemistry, ecology, links with urban and agricultural practices, sociologic impact and legal treatment). This joint scientific appraisal (a so-called French ESCo "**E**xpertise **S**cientifique **Co**llective") should help the public authorities to redefine French regulatory texts dealing with the complex and controversial issue of eutrophication, especially the role played by nitrogenous and phosphorous nutrients in this phenomenon. This will better establish the revision of vulnerable zones in respect of the Nitrate Directive (Directive 91/676/EEC) as well as the implementation of the European directives on the management of aquatic environments, i.e. the Water Framework Directive (WFD, Directive 2000/60/EC), the Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC) and the Urban Waste Water Treatment Directive (UWWTD, 91/271/EEC).

The global appraisal is based on approximately 4000 scientific references. The present subset, dealing only with the modelling of marine eutrophication, is based on 291 references. They result from a first extraction from the Web of Science Core Collection thanks to a bibliographic search equation, followed by the discard of some inappropriate references (e.g. dealing with inland waters) and the addition of some pertinent scientific papers known by the authors, but not selected by the search equation. As the work has been split into four distinct domains (estuaries, lagoons, coastal seas, "green tides"), Table 1 gives for each domain the search equation and the number of papers selected by it, along with the numbers of discarded and added papers. The number of published modelling studies per year (Fig. 1) reveals that the modelling of marine eutrophication is relatively recent, beginning at the end of the 80's. Whereas studies on estuaries and coastal areas had shown a regular increase since 1990, studies on "green tides" as well as on lagoons have kept a weak steady state since the mid 90's. These research papers were published in 76 journals, the 5 most cited being Ecological Modelling (15.8%), Journal of Marine Systems (13.6%), Estuarine Coastal and Shelf Science (9.7%), Marine Ecology Progress Series (5.4%), Hydrobiologia (3.9%), collectively publishing 49.4% of all papers.

3. Aims of the modelling approach

The first aim of the eutrophication models was to help understanding the eutrophication process and to reproduce its main features, i.e. the intense phytoplanktonic blooms (Chan et al., 2002; Kishi & Ikeda, 1986; Dippner, 1993; Baretta et al., 1994; Fennel, 1995; Tett & Walne, 1995; Yanagi et al., 1995; McEwan et al., 1998; Cugier et al., 2005; Tamvakis et al., 2012) or the mass accumulation of ulvaceae (Ménesguen & Salomon, 1988; Bendoricchio et al., 1994; Coffaro & Sfriso, 1997; Martins & Marques, 2002; Lovato et al., 2013; Coffaro & Bocci, 1997; Brush & Nixon, 2010; Trancoso et al., 2005; Silva-Santos et al., 2006), and the possible hypoxia (or anoxia) induced in the bottom waters (Chapelle, 1995; Humborg et al., 2000; Chapelle et al., 1994; Oguz et al., 2000; Grégoire & Lacroix, 2001; Karim et al., 2002; Tuchkovenko & Lonin, 2003; Grégoire et al., 2008; Grégoire & Soetaert, 2010; Evans & Scavia, 2011; Pena et al., 2010; Wan et al., 2012; Große et al., 2017). Because the dynamics of estuaries is strongly controlled by physical drivers (river flow rate, turbidity, haline stratification), the estuarine models have very often been used to assess the respective role of these drivers, especially on the onset of hypoxic conditions (Peterson & Festa, 1984; Cole & Cloern, 1987; Soetaert et al., 1994; Vanderborght et al., 2002; Muylaert et al., 2005; Talke et al., 2009; Nash et al., 2011; Gypens et al., 2013; Gallegos, 2014; O'Boyle et al., 2015; Mathews et al., 2015; Fear et al., 2004; Talke et al., 2009; Robson et al., 2008; Talke et al., 2009; Nash et al., 2011; Arndt et al., 2011; Wang et al., 2013; Liu & de Swart, 2015; O'Boyle et al., 2015; Chen et al., 2014; Chen et al., 2015; Benoit et al., 2006; Bruce et al., 2011b; Hipsey et al., 2013; Cho et al., 2015; Miguez et al., 2001).

Beyond these main aspects, the models have been also used to explore the competition between diverse primary producers: diatoms vs dinoflagellates or Prymnesiophyceae (*Phaeocystis* sp.) as well as diatoms vs cyanobacteria (Petihakis et al., 1999; Guillaud & Ménesguen, 1998; Guillaud et al., 2000; Gypens et al., 2007; van den Berg et al., 1996a; Blauw et al., 2009; Spatharis & Tsirtsis, 2013), *Ulva* vs phanerogams *Zostera* and *Ruppia* (Giusti & Marsili-Libelli, 2005; Cioffi & Gallerano, 2006; Canal-Verges et al., 2014), or macro- vs microalgae (Baird et al., 2003; Madden & Kemp, 1996; Buzzelli et al., 2014; Sohma et al., 2004).

Table 1
Statistics of the reference collecting process.

	Search equation	References collected	References discarded	References added	References used
Estuaries	TITLE: (eutrophic* OR oxygen* OR hypox* OR anox* OR phytoplankton* OR "harmful alga"* OR nutrient* OR cyanobact* OR proliferat*) AND (model* OR numerical OR simulat* OR predict*) AND estuar*	129	51	5	83
Lagoons	TITLE: (((eutrophic* OR oxygen* OR hypox* OR anox* OR phytoplankton* OR "harmful alga"* OR nutrient* OR cyanobact* OR proliferat*) AND (model* OR numerical OR simulat*) AND lagoon*))	38	13	5	30
Coastal seas	TITLE: ((eutrophic* OR oxygen* OR hypox* OR anox* OR phytoplankton* OR "harmful alga"* OR phaeocystis OR nutrient* OR cyanobact* OR proliferat*) AND (model* OR numerical OR simulat* OR predict*) AND (marine OR bay* OR coastal OR sea)) AND TOPIC: (eutrophic*)	184	68	37	153
"Green tides"	TITLE: ((("green alga"* OR "green tide*" OR macroalg* OR ulva) AND (model* OR numerical OR simulat*))) AND TOPIC: (eutrophic* OR hypox* OR anox*)	17	4	12	25

Rapidly, the modelling effort has been applied to assess the respective role of various nutrient sources, mainly the continental watersheds vs the oceanic inputs (Lowery, 1998; Meeuwig et al., 2000; Kauppila et al., 2003; Arismendez et al., 2009; Plus et al., 2006; Malhadas et al., 2014; Timmermann et al., 2010; Beck et al., 2001; Roelke et al., 1999), or the upward flux from sediments (Fernandez et al., 2012; van Raaphorst et al., 1988; Magalhaes et al., 2008; Liu et al., 2009; Bruce et al., 2011a). The inputs of nitrogen from the atmosphere have been recently taken into account in the Baltic Sea (Ruoho-Airola et al., 2012), the North Sea (Djambazov & Pericleous, 2015; Dulière et al., 2017; Troost et al., 2013), and in Singapore region (Sundarambal et al., 2010); they can reach up to 29% of the nitrogen inputs in Belgian offshore waters (Dulière et al., 2017), but they remain in most cases of minor importance in the case of strong coastal eutrophication. Since the 2000's, the use of numerical tracers has become more and more popular to track the fate of various nutrient loadings, either considered as simple passive and conservative dyes (Kasih & Kitada, 2004; Lacroix et al., 2007; Wild-Allen et al., 2010; Eilola et al., 2014; Djambazov & Pericleous, 2015) or as active and form-changing variables (Ménesguen et al., 2006; Sugimoto et al., 2010; Timmermann et al., 2010; Troost et al., 2013; Los et al., 2014; Dulière et al., 2017; Große et al., 2017). Forcing models with time-series of past loadings and meteorology has allowed reconstruction of the decadal change of the eutrophication status of large areas, such as the North Sea (Pätsch & Radach, 1997; de Vries et al., 1998; Lancelot et al., 2009), the Gulf of Mexico (Justic et al., 2002; Justic et al., 2003; Scavia et al., 2003; Justic et al., 2007; Evans & Scavia, 2011; Obenour et al., 2015; Feist et al., 2016), the Baltic Sea (Neumann & Schernewski, 2008). As one of the recurrent questions asked by the decision-makers is about the limiting factor of the excessive algal production, several studies have been focused on that question (Gallegos & Jordan, 1997; Fan & Song, 2014; Liu et al., 2014; O'Boyle et al., 2015; Lin et al., 2008; Ménesguen & Salomon, 1988; Ménesguen, 1992). However, the commonest use of the models is the simulation of scenarios for operational recommendations. Particularly in the case of strongly modified ecosystems, far from their natural status and from the domain of linear response to nutrient loadings, models are the only way to forecast the effects of various scenarios of loading reduction. The current situation of loadings is taken as the 100% reference, and reductions (<100%) or augmentations (>100%) can be tested (Olufeagba & Flake, 1981; Lonin & Tuchkovenko, 2001; Onandia et al., 2015; Lung, 1988; Lung & Testerman, 1989; Meeuwig, 1999; Lung & Nice, 2007; Camacho et al., 2015; Garcia et al., 2002; Sohma et al., 2008; Lenhart et al., 2010), Lacroix et al., 2007, (Hipsey et al., 2013; Skerratt et al., 2013; Liu & Chan, 2014; del Barrio et al., 2014; Timmermann et al., 2014; Steward & Lowe, 2010; Saraiva et al., 2007; Gypens et al., 2013; Murray & Parslow, 1999; Cugier et al., 2005; Cerco & Noel, 2013; Qiao et al., 2014; Los et al., 2014; Justic et al., 2007; Scavia & Donnelly, 2007; Liu & Scavia, 2010; Testa et al., 2014). More recently, models have been used to assess the ecological status of the natural ecosystems without any anthropogenic impact, the so-called "pristine" situation (Lancelot et al., 2009; Lancelot et al., 2014). Simulations with current nutrient loadings, reduced ones, or

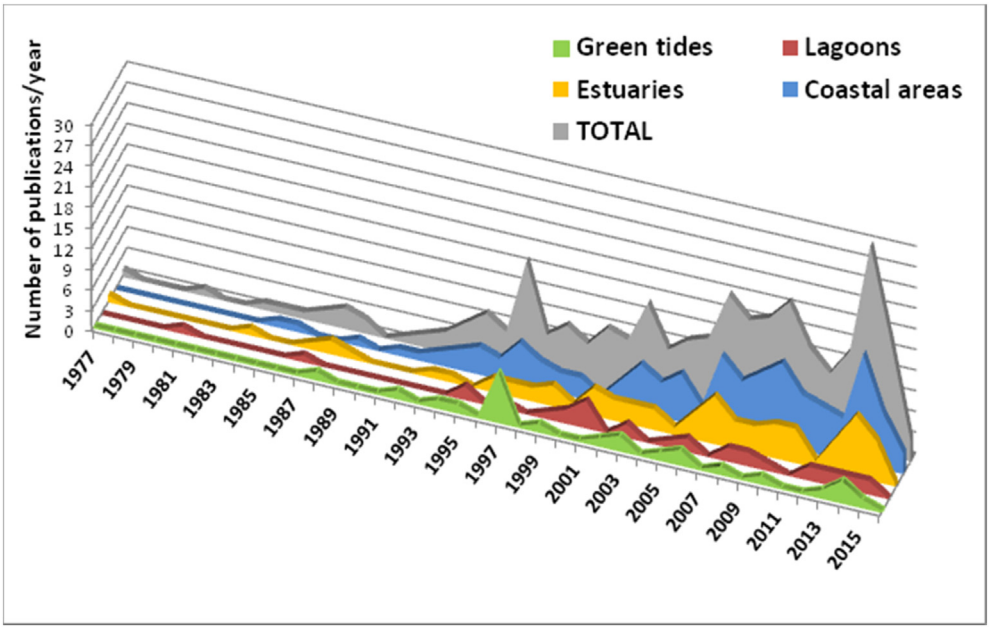

Fig. 1. Number of modelling studies of marine eutrophication over time.

even pristine ones have been compared, especially in the case of wide coastal areas, to the requirements of the so-called Good Ecological Status (GES), in compliance with the European Commission Directives (WFD and MSFD) or the OSPAR and HELCOM requirements (Nobre et al., 2005; Allen et al., 2008; Troost et al., 2014). Up to now, very few attempts have been made to invert the cause-effect relationship linking nutrient loadings and marine eutrophication status, in order to be able to compute directly the reductions of loadings necessary to ensure a fixed level of marine ecological status (Borsuk et al., 2001a; Borsuk et al., 2004). Recently, this question has been treated thanks to the Simplex optimization technique, provided the cause-effect relationship is not too far from linearity. This allows computing the most cost-effective terrestrial load reductions which could lead to the GES (Los et al., 2014; Ménesguen et al., 2018). In some cases, models have also been used to explain the lack of severe eutrophication in coastal areas heavily enriched by the tributaries (LePape & Ménesguen, 1997; Powley et al., 2014; Machado & Imberger, 2014).

Especially in lagoons, some models have been used to assess the benefit which could be expected from various remediation techniques: deriving tributaries from the lagoon into the coastal sea (Ren et al., 2014), harvesting the algae (Cioffi & Gallerano, 2001; Allegretto & Papini, 2008), increasing the water turn-over (Cioffi et al., 1995; Lonin & Tuchkovenko, 2001; Onandia et al., 2015), reducing the oyster biomass under cultivation (Chapelle et al., 2001), or removing the organic-rich surficial layer of the sediment (Cioffi & Gallerano, 2001). The impact of eutrophication on shellfish or fish farming has been explored (Bacher et al., 1997; Grangeré et al., 2010; Timmermann et al., 2014), as well as the inverse link, i.e. the possible use of cultivated herbivorous to limit the phytoplankton proliferation (Fulford et al., 2007; Cerco et al., 2010). This is a very applied form of the more general question about the modifications of the entire foodweb by the eutrophication process (Doi & Nitta, 1991; Lancelot et al., 2002; Barausse et al., 2009; Townsend, 2014; Luang et al., 2014). Very few models have investigated the possible interactions of eutrophication and toxic pollutants (Driscoll et al., 2012; Legovic, 1997).

Up to now, the economic impact of the marine eutrophication remains rarely modelled (Cellina et al., 2003; Lancelot et al., 2011; Shepherd et al., 2007; Laukkanen et al., 2009; Ahlvik et al., 2014; Timmermann et al., 2014). The probable impact of climate change on the marine eutrophication is a growing issue (Martins et al., 2008; Eilola et al., 2012; Meier et al., 2011; Meier et al., 2012; Skogen et al., 2014; Bring et al., 2015; Lessin et al., 2014; Lopes et al., 2015; Ryabchenko et al., 2016; Picart et al., 2015). However, turning the results of models into a quantitative, distributed risk index which can be mapped geographically remains unfamiliar for eutrophication modellers (Druon et al., 2004; Cosme et al., 2015).

4. Target sites and periods covered by the models

The global geographic distribution of the published models of marine eutrophication is given in Fig. 2. Historically speaking, the modelling effort has pointed out the hotspots of marine eutrophication.

Among the coastal areas threatened by eutrophication, the top five which have given rise to the biggest modelling efforts are:

1/ The Baltic Sea (Fennel, 1995; Fennel, 2010; Carlsson et al., 1999; Helminen et al., 1998; Neumann et al., 2002; Berger et al., 2004); (Neumann & Schernewski, 2005; Savchuk & Volkova, 1990; Savchuk & Wulff, 1999; Korpinen et al., 2004; Savchuk & Wulff, 2007; Neumann & Schernewski, 2008; Savchuk & Wulff, 2009; Laukkanen et al., 2009; Eilola et al., 2009; Eilola et al., 2011; Yekta & Rahm, 2011; Meier et al., 2011; Meier et al., 2012; Wan et al., 2012; Eilola et al., 2012; Bergstrom et al., 2013; Ahlvik et al., 2014; Eilola et al., 2014; Lehmann et al., 2014; Lessin et al., 2014; Dimberg & Bryhn, 2014; Almroth-Rosell et al., 2015; Janssen et al., 2015; Ryabchenko et al., 2016; Almroth & Skogen, 2010); (Skogen et al., 2014; Edelvang et al., 2005); (Bring et al., 2015).

2/ The North Sea (Baretta et al., 1994; Slomp et al., 1996); (van den Berg et al., 1996a; van den Berg et al., 1996b; Baretta-Bekker et al., 1997; Pätsch & Radach, 1997; Tett & Walne, 1995; de Vries et al., 1998;

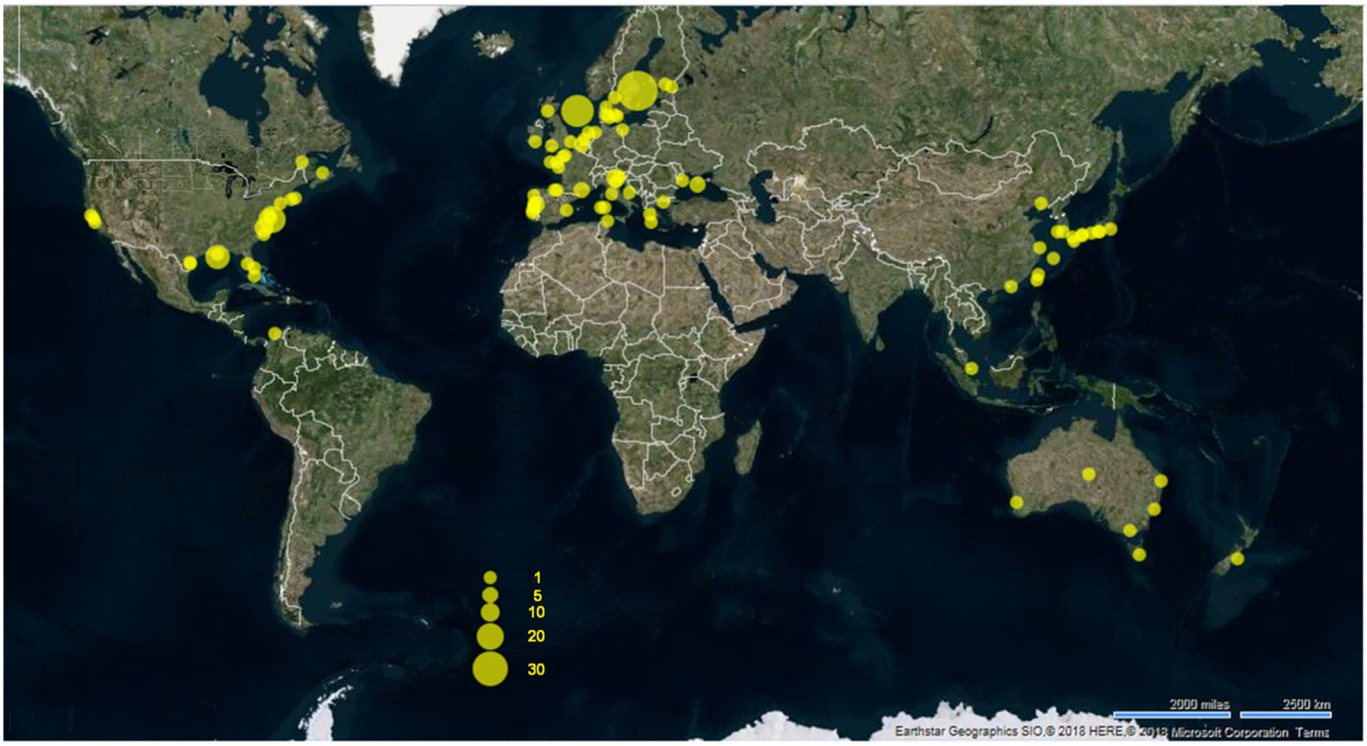

Fig. 2. World distribution of the models of marine eutrophication.

Druon et al., 2004; Lancelot et al., 2004; Lancelot et al., 2005; Lancelot et al., 2007; Gypens et al., 2007; Lacroix et al., 2007; Allen et al., 2008; Gypens et al., 2009; Lancelot et al., 2009; Blauw et al., 2009; Lenhart et al., 2010; Lancelot et al., 2011; Troost et al., 2013; Passy et al., 2013); (Troost et al., 2014; Los et al., 2014; Lancelot et al., 2014; Picart et al., 2015; Große et al., 2017).

3/ The Gulf of Mexico (Chen et al., 1997; Justic et al., 2007; Morse & Eldridge, 2007; Hetland & DiMarco, 2008; Evans & Scavia, 2011; Obenour et al., 2015; Feist et al., 2016).

4/ The Chesapeake Bay (Cerco & Cole, 1993; Cerco, 1995; Fulford et al., 2007; Cerco et al., 2010; Liu & Scavia, 2010; Cerco & Noel, 2013; Sturdivant et al., 2013; Muller & Muller, 2015).

5/ The Black Sea (Yakushev & Neretin, 1997; Legovic, 1997; Oguz et al., 2000; Grégoire & Lacroix, 2001; Lancelot et al., 2002; Grégoire et al., 2008; Grégoire & Soetaert, 2010).

In their review of hypoxia models, (Pena et al., 2010) add to these five big sites three others, the Tokyo Bay, the Gulf of St. Lawrence and the Cariaco Basin in the north of Venezuela. A lot of smaller sites have also been modelled, from France (Chapelle et al., 1994; Guillaud & Ménesguen, 1998; Pénard et al., 2007; Guillaud et al., 2000; Cugier et al., 2005; Passy et al., 2016) or Italy (Dippner, 1993; Druon et al., 2004; Barausse et al., 2009) to Japan (Kishi & Ikeda, 1986; Yanagi et al., 1995; Doi & Nitta, 1991; Karim et al., 2003a; Mukai et al., 1985; Karim et al., 2002; Karim et al., 2003b; Kasih & Kitada, 2004; Sohma et al., 2004; Sugimoto et al., 2010; Koriyama et al., 2011), China (Chau, 2007; Li et al., 2008; Qiao et al., 2014) or Australia (Machado & Imberger, 2014; McEwan et al., 1998; Murray & Parslow, 1999; Wild-Allen et al., 2010).

Regarding the green macroalgae, the first model appeared in Brittany (France) at the end of the 80's (Ménesguen & Salomon, 1988), followed by improved versions up to now (Ménesguen, 1992; Ménesguen et al., 2006; Perrot et al., 2014). The biggest “green tide” in the world at the end of the 20th century, the Venice lagoon, has been modelled mainly at the end of the 90's (Solidoro et al., 1995; Solidoro et al., 1997b; Coffaro & Sfriso, 1997; Coffaro & Bocci, 1997; Bergamasco & Zago, 1999; Lovato et al., 2013), whereas the Sacca di Goro lagoon, south of the Po delta, waited for the early 2000's (Cellina et al., 2003; Bondavalli, 2003). The present biggest “green tide”, native from the seaweed farms of the Shandong coast and drifting in the Yellow Sea up to Qingdao beaches, begun to be modelled very recently (Zhang et al., 2015). Besides these main sites, much smaller “green tides” have been studied in Portugal (Martins et al., 1997; Martins & Marques, 2002; Silva-Santos et al., 2006; Martins et al., 2008; Trancoso et al., 2005), in Denmark (Nielsen, 1997; Canal-Verges et al., 2014), and in the U.S.A (Brush & Nixon, 2010).

In addition to the cases of green macroalgae proliferation previously reported, several Mediterranean lagoons have been modelled for their hypoxic episodes or their excessive phytoplanktonic blooms: the French Thau lagoon (Chapelle, 1995; Bacher et al., 1997; Chapelle et al., 2000; Chapelle et al., 2001; Plus et al., 2003; Plus et al., 2006), the Italian lagoons of Venice (Dejak et al., 1987), Orbetello (Cioffi et al., 1995; Giusti & Marsili-Libelli, 2005; Rubegni et al., 2013), Fogliano (Cioffi & Gallerano, 2001; Hull et al., 2008), Tortoli (Cioffi & Gallerano, 2006) and Cabras (Padedda et al., 2010), the Spanish lagoon of Albufera de Valencia (Fernandez et al., 2012; Onandia et al., 2015), and the Greek ones of Vistonis (Gikas et al., 2009), Gialova (Petihakis et al., 1999), Aitoliko-Messolonghi (Gianni & Zacharias, 2012). Few lagoons have been modelled along the Atlantic coasts: the Spanish Victoria lagoon in the Bay of Biscay (Zouiten et al., 2013), the Portuguese lagoons Obidos (Malhadas et al., 2014) and Ria de Aveiro (Lopes et al., 2015), the Columbian Ciénaga de Tesca lagoon (Lonin & Tuchkovenko, 2001), and the USA lagoons located in Texas: San Antonio Bay and Copano Bay (Turner et al., 2014) or in the New-York Bight (Olufeagba & Flake, 1981).

More often than all the other coastal ecosystems, estuaries have been modelled by groups and compared: 6 in the U.S.A (Cole & Cloern, 1987), 10 in the Gulf of Mexico (Lowery, 1998), 12 in Florida (Steward & Lowe, 2010), 15 on the Canadian Prince Edwards island (Meeuwig, 1999), 34 in the world (Borsuk et al., 2001a), 19 Finnish estuaries around the Baltic Sea (Meeuwig et al., 2000; Kauppila et al., 2003), 9 in Portugal (Saraiva et al., 2007), 90 in Ireland (O'Boyle et al., 2015). Some big estuaries have concentrated a lot of studies: the Chesapeake Bay (Lung, 1988; Lung & Testerman, 1989; Madden & Kemp, 1996; Shen, 2006; Adamack et al., 2012; Lung & Nice, 2007; Gallegos, 2014; Gallegos & Jordan, 1997), the Neuse river in North Carolina (Lung, 1988; Borsuk et al., 2002; Borsuk et al., 2001b; Borsuk et al., 2004; Fear et al., 2004; Millie et al., 2006), the Saint Lawrence estuary (Benoit et al., 2006; Katsev et al., 2007), the Yang-Tsé-Kiang estuary (Wang et al., 2013; Chen et al., 2014; Fan & Song, 2014; Liu et al., 2014; Chen et al., 2015), the Belgian Scheldt estuary (Soetaert et al., 1994; Vanderborght et al., 2002; Muylaert et al., 2005; Arndt et al., 2011; Gypens et al., 2013), the Portuguese Mondego estuary (Patricio & Marques, 2006; Magalhaes et al., 2008; Baeta et al., 2011) and Tagus estuary (Alvera-Azcarate et al., 2003).

The duration of the simulations covers a few months to a year in the oldest studies, whereas more and more studies since the 2000's deal with 10-years simulations (Plus et al., 2006) or more (Humborg et al., 2000; Lancelot et al., 2007; Blauw et al., 2009; Große et al., 2017).

5. Modelling tools

- Deterministic versus statistical models

Quantitative descriptors of eutrophication status can be computed thanks to either statistical models using measurements of causative variables as inputs and empirically calibrated on some data set, or deterministic models, based on a mathematical theory of the eutrophication process, expressed in a set of differential equations needing some physical/biogeochemical parameters. If the statistical approach was favored before 2000 (Cole & Cloern, 1987; Gowen et al., 1992; Lowery, 1998) owing to its modest computation requirements, the deterministic approach has become the main one since the 90's, thanks to the tremendous increase of computing power and accessibility. However, the statistical approach has never been abandoned, especially in the U.S.A, and the increase of data availability may have even caused a renewed interest, either within the classical multi-regressive frame or within a Bayesian approach producing probability distributions of expected final descriptors under various scenarios, thereby providing some estimate of the uncertainty. Multiregressive models have been used by (Carlsson et al., 1999; Meeuwig, 1999; Meeuwig et al., 2000; Justic et al., 2002; Scavia et al., 2003; Kauppila et al., 2003; Turner et al., 2005; Silva-Santos et al., 2006; Muylaert et al., 2005; Millie et al., 2006; Hetland & DiMarco, 2008; Melesse et al., 2008; Liu & Scavia, 2010; Koriyama et al., 2011; Chang et al., 2012; Tamvakis et al., 2012; Cerco & Noel, 2013; Gallegos, 2014; Chen et al., 2014; Timmermann et al., 2014; Mathews et al., 2015). Bayesian models can be found in (Arhonditsis et al., 2007; Borsuk et al., 2001b; Borsuk et al., 2002; Borsuk et al., 2004; Stow & Scavia, 2009; Liu & Scavia, 2010; Obenour et al., 2015). Whereas statistical models provide an estimate of the uncertainty but are bound to the system in which they were empirically developed (even if Bayesian models may be more adaptable), deterministic models may be run under different constraints in other systems, but the uncertainty associated with their results is difficult to assess.

- Spatial frame

As presented in Fig. 3, the spatial frame of the model can go from none, in the so-called 0D models (Howarth et al., 1999; Kimmerer et al., 1993; Rosenberg et al., 1977) up to a full-3D grid (Solidoro et al., 1997a); favorite intermediate frames are the horizontal 1D grid (for

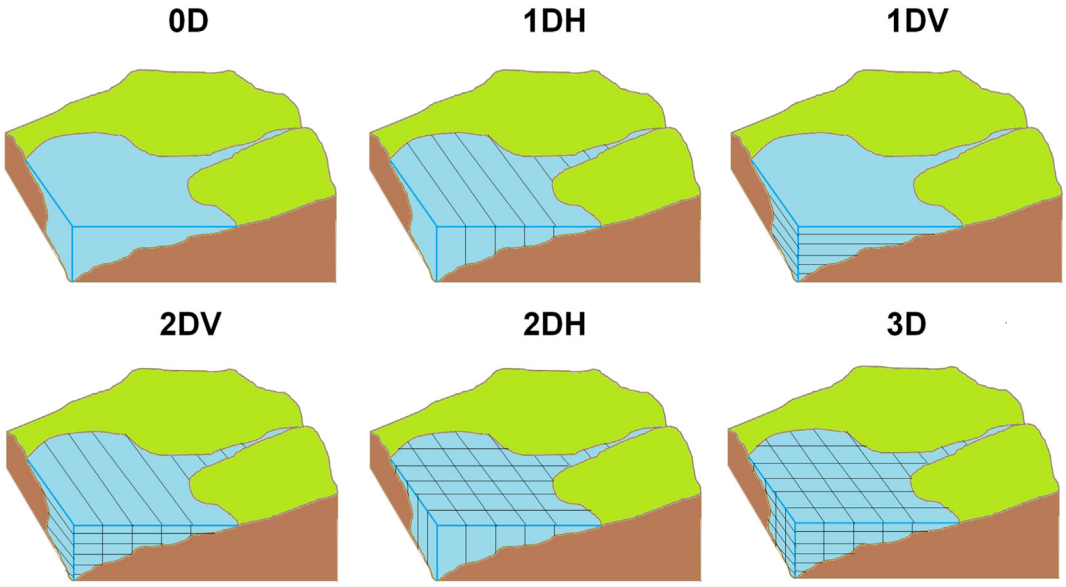

Fig. 3. Diagrams illustrating various spatial approaches of a marine coastal water body with an estuary.

estuaries), vertical 1D grid (for lagoons and enclosed seas), horizontal 2D grid (for coastal shallow waters without strong freshwater influence), vertical 2D grid (for strongly stratified estuarine and river plume systems).

Various shapes of the grid meshes have been used (Fig. 4). On the horizontal dimension, about 68% of the 2D and 3D models use Cartesian, rectangular meshes, 27% use orthogonal curvilinear grids, whereas only 5% use finite triangular elements. On the vertical dimension, the models usually use layers of unequal thickness, in order to better simulate the sub-surface or the near-bottom gradients. In 2DV (Fig. 3) and full-3D models, many models use a relative vertical discretization (with a so-called σ–coordinate) instead of a horizontally uniform, absolute vertical discretization (with a so-called z–coordinate), as shown in Fig. 5.

- Biogeochemical/biological components

Surprisingly, some models of marine eutrophication do not take explicitly into account the nutrients, which are the necessary fuel of excessive biomass production. Several estuarine models are purely physically driven (Peterson & Festa, 1984; Miguez et al., 2001; Li et al., 2003; Fear et al., 2004; Hagy & Murrell, 2007; Steward & Lowe, 2010; Das et al., 2010; Wang et al., 2013; O'Boyle et al., 2015; Beck et al., 2001; Cho et al., 2015; Talke et al., 2009) as well as some coastal zone models (Smith & Hollibaugh, 1989; Legovic, 1997; Helminen et al., 1998; Druon et al., 2004; Pénard et al., 2007; Wan et al., 2012; Chang et al., 2012; Spatharis & Tsirtsis, 2013). "Green tide" models only interested in deleterious effects, i.e. drift and accumulation of free-floating algae (Canal-Verges et al., 2014) or bottom oxygen depletion (Cellina et al., 2003) can limit the biological component of their model to an algal growth rate depending on temperature. A few models of coastal hypoxic zones, especially in the Gulf of Mexico, are still based on the fundamental couple of state variables characteristic of the first models of river water quality, i.e. the Biological Oxygen Demand (BOD) and the Dissolved Oxygen (DO) (Carlsson et al., 1999; Liu & Scavia, 2010; Liu et al., 2010; Evans & Scavia, 2011). (Lehmann et al., 2014).

However, the vast majority of deterministic eutrophication models include a more or less comprehensive and detailed description of the potentially limiting nutrients. In the 90s, most models used only DO (eventually BOD) along with four state variables (NPZD) summarizing

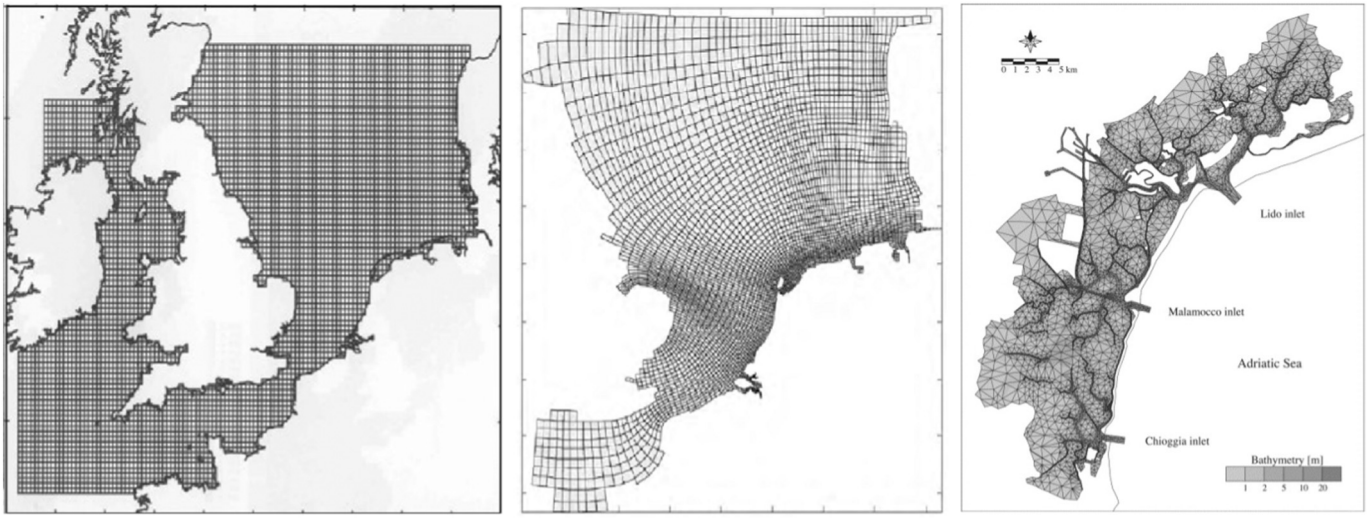

Fig. 4. Cartesian vs curvilinear vs finite triangular element grids (sources: RBINS/Belgium, Deltares/The Netherlands, ISMAR-CNR/Italy).

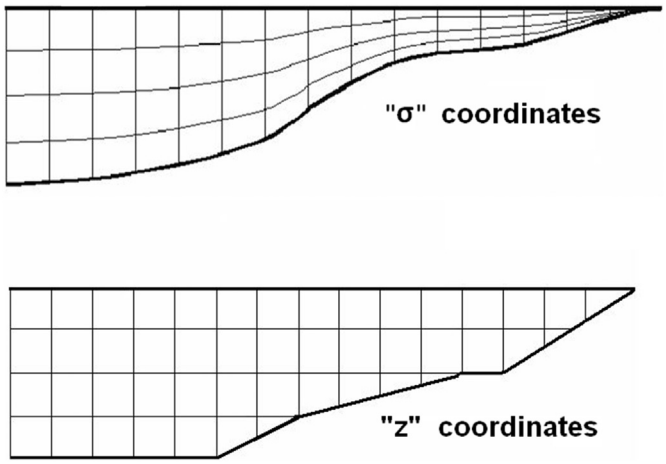

Fig. 5. σ–coordinate vs z–coordinate in the vertical plane.

the biogeochemical cycle of the most limiting nutrient: N for nutrient, P for phytoplankton, Z for zooplankton and D for detritus (van Raaphorst et al., 1988; Fennel, 1995; Tett & Walne, 1995; Bergamasco & Zago, 1999). As nitrogen is considered as the first limiting nutrient in marine ecosystems, especially in summer when the risk of anoxia is greater, a lot of models use nitrogen only (Gowen et al., 1992; Bergamasco & Zago, 1999; Trancoso et al., 2005; Solidoro et al., 1995; Coffaro & Bocci, 1997; Bacher et al., 1997; Baird et al., 2003; Vanderborght et al., 2002; Plus et al., 2003; Plus et al., 2006; Turner et al., 2014; Giusti & Marsili-Libelli, 2005; Arismendez et al., 2009; Dugdale et al., 2013; del Barrio et al., 2014; Bruce et al., 2011a; Liu & de Swart, 2015; Bruce et al., 2011b; Gowen et al., 1992; Fennel, 1995; Tett & Walne, 1995; Guillaud & Ménesguen, 1998; Edwards et al., 2003; Nobre et al., 2005; Sugimoto et al., 2010; Machado & Imberger, 2014; Oguz et al., 2000; Grégoire & Lacroix, 2001; Sugimoto et al., 2010). More and more models simulate the phosphorus cycle in parallel with the nitrogen one (Ménesguen & Salomon, 1988; Ménesguen, 1992; Bendoricchio et al., 1994; Solidoro et al., 1995; Coffaro & Sfriso, 1997; Perrot et al., 2014; Lovato et al., 2013; Martins & Marques, 2002; Brush & Nixon, 2010; Martins et al., 2008; Dejak et al., 1987; Humborg et al., 2000; Lonin & Tuchkovenko, 2001; Schernewski & Wielgat, 2001; Rasmussen et al., 2009; Gikas et al., 2009; Padedda et al., 2010; Malhadas et al., 2014; Petihakis et al., 1999; Lopes et al., 2015; Chapelle, 1995; Zouiten et al., 2013; Shen, 2006; Robson et al., 2008; Timmermann et al., 2010; Liu et al., 2014; Gallegos & Jordan, 1997; Roelke et al., 1999; Saraiva et al., 2007; Chan et al., 2002; Buzzelli et al., 2014; Nash et al., 2011; Lung & Nice, 2007; Liu et al., 2009; Liu & Chan, 2014; Camacho et al., 2015; Kishi & Ikeda, 1986; Savchuk & Wulff, 1999; Korpinen et al., 2004; Qiao et al., 2014; Edelvang et al., 2005; Neumann & Schernewski, 2005; Bierman et al., 1994; Wild-Allen et al., 2010; McEwan et al., 1998; Guillaud et al., 2000; Karim et al., 2002; Karim et al., 2003a; Karim et al., 2003b); Tuchkovenko & Lonin, 2003; Chau, 2007; Neumann & Schernewski, 2008; Blauw et al., 2009); (Eilola et al., 2012; Meier et al., 2012; Wan et al., 2012; Troost et al., 2013; Troost et al., 2014; Lessin et al., 2014; Almroth-Rosell et al., 2015; Janssen et al., 2015). Especially in estuaries, a lot of statistical models also use N- and P loadings as causative variables in order to better constrain the systems where there may be a seasonal alternance in N and P limitation or a co-limitation of nutrients (Meeuwig, 1999; Meeuwig et al., 2000; Cole & Cloern, 1987; Muylaert et al., 2005; Gallegos, 2014; Timmermann et al., 2014; Mathews et al., 2015; Millie et al., 2006; Lowery, 1998; Kauppila et al., 2003; Chen et al., 2014; Tamvakis et al., 2012; Melesse et al., 2008; Muller & Muller, 2015). The silicon cycle is added in the cases where eutrophication is dominated by phytoplankton production, in order to model the dynamics of diatoms or the export/retention of silica (Soetaert et al., 1994; LePape & Ménesguen, 1997; Grangeré et al., 2010; Arndt et al., 2011; Gypens et al., 2013; Baretta et al., 1994; van den Berg et al., 1996a; Baretta-Bekker et al., 1997; Pätsch & Radach, 1997; de Vries et al., 1998; Kasih & Kitada, 2004; Cugier et al., 2005; Li et al., 2008; Lacroix et al., 2007; Lancelot et al., 2005; Lancelot et al., 2007; Lancelot et al., 2009; Savchuk & Wulff, 2009; Lancelot et al., 2011; Lancelot et al., 2014; Gypens et al., 2007; Gypens et al., 2009; Blauw et al., 2009; Passy et al., 2013; Ryabchenko et al., 2016; Passy et al., 2016; Murray & Parslow, 1999; Powley et al., 2014; Chapelle et al., 1994; van den Berg et al., 1996b; Eilola et al., 2009; Murray & Parslow, 1999; Lancelot et al., 2002; Cerco & Cole, 1993; Park et al., 2005; Sohma et al., 2004; Große et al., 2017). In order to allow varying stoichiometric composition of some compartments, some models use carbon units for zooplankton biomass and detrital variables (Maar et al., 2010; Sohma et al., 2008; Magalhaes et al., 2008; Hipsey et al., 2013; Chen et al., 2015; Lin et al., 2008; Blauw et al., 2009; Große et al., 2017). Some simulate explicitly the biological uptake/production of dissolved inorganic carbon (Flindt & Kamp-Nielsen, 1997; Grégoire et al., 2008; Grégoire & Soetaert, 2010; Skerratt et al., 2013; Wild-Allen & Rayner, 2014; Allen et al., 2008; Picart et al., 2015; Feist et al., 2016), but only one (Vanderborght et al., 2002) simulates the complete carbon cycle, including the carbonate/bicarbonate buffer. So, up to now, eutrophication modelling has not been really coupled to the pH simulation necessary to assess a possible interaction with ocean acidification.

The biological state variables of almost all these models are bulk variables (e.g. total phytoplankton, or diatoms and dinoflagellates, or pelagic diatoms, epibenthic diatoms and macrophytes (Madden & Kemp, 1996) or pre-determined target species with literature-derived physiological parameters: *Phaeocystis* (Gypens et al., 2007; Lacroix et al., 2007; Blauw et al., 2009; Lancelot et al., 2014), *Ulva* (Ménesguen & Salomon, 1988; Ménesguen, 1992; Bendoricchio et al., 1994; Solidoro et al., 1995; Coffaro & Sfriso, 1997; Perrot et al., 2014; Lovato et al., 2013), *Enteromorpha* (Martins & Marques, 2002; Martins et al., 2008), *Ulva* and *Gracilaria* (Brush & Nixon, 2010). Trying to simulate deterministically the emergence of the "pristine" biodiversity in non-eutrophicated systems and its modification under increasing nutrient enrichment remains uncommon (Coffaro et al., 1997; Nielsen, 1997), perhaps because of the too difficult validation and the too weak improvement of the result reliability relatively to the more classical approaches, including simple multiple regression models (Silva-Santos et al., 2006; Arhonditsis et al., 2007; Justic et al., 2002; Liu & Scavia, 2010; Bergstrom et al., 2013; Carlsson et al., 1999).

As settling of dead organic matter triggers anoxic, reducing conditions in the sediment, which can modulate the nutrient exchanges between the bottom water and the sediment, particularly in confined lagoons or inland seas, some models have included an explicit sedimentary part to the nutrient cycles (Petihakis et al., 1999; Baretta et al., 1994; Cerco & Cole, 1993; Cerco et al., 2010; Benoit et al., 2006), completed with sulfur, iron and manganese biogeochemical cycles (Chapelle et al., 2001; Sohma et al., 2004; Soetaert & Middelburg, 2009; Testa et al., 2014; Slomp et al., 1996; Morse & Eldridge, 2007; Yekta & Rahm, 2011; Katsev et al., 2007; Sohma et al., 2008; Fennel et al., 2009; Skerratt et al., 2013; Yakushev et al., 2007; Testa et al., 2014).

The consumer levels of the ecosystem (herbivorous, carnivorous) are included in models focused on the impact of eutrophication upon aquaculture and fisheries (Sturdivant et al., 2013; Fulford et al., 2007; Grégoire et al., 2008; Luang et al., 2014; Fennel, 2010; Doi & Nitta, 1991). Such studies use the end-to-end modelling approach, often with the Ecopath-Ecosim formulation of the trophic web (Lin et al., 2007; Baeta et al., 2011; Patricio & Marques, 2006; Timmermann et al., 2014; Barausse et al., 2009; Cerco et al., 2010; Townsend, 2014).

6. Reliability of models: how do models relate to observations?

The first task of every model is to mimic some observed key-descriptors of the eutrophication process, i.e. to compare reasonably

well to measurements made in the past. This is done thanks to the calibration phase, which consists in tuning the parameters and the boundary conditions of the equations of a deterministic model, or computing the least square estimates of the coefficients of a multi-regressive model. In case of deterministic models, automatic calibration procedures (Gauss-Newton steepest descent method, Nelder-Mead nonlinear simplex, genetic algorithm…) have been rarely used for the simple eutrophication box-models containing very few state variables (Giusti & Marsili-Libelli, 2005; Shen & Kuo, 1996; Shen & Kuo, 1998; Shen, 2006; Grégoire et al., 2008). They have become unusable in current fine 3D models exhibiting hundreds of state variables. Parameter calibration remains massively a handmade process, by trial and error, and suffers from the fact that the state variables can be unequally constrained by observed data. Whereas remote sensing from satellites now provides extended coverage of some surface state variables (temperature, suspended particulate matter, chlorophyll *a*), and some buoys can deliver in real time high frequency measurements of chlorophyll *a* and dissolved nutrients or oxygen, about half of the state variables of marine eutrophication models remain hidden by lack of regular measurement: detrital forms of N, P or Si, adsorbed phosphate…
The simplest metrics used to quantify the adequacy between models and observations are based on sums of instantaneous discrepancy between the measured and simulated values, performed upon their algebraic value (bias), or their absolute value (gap) or frequently their quadratic value, with subsequent extraction of the square root (distance, called the Root Mean Square Error RMSE). However, more sophisticated metrics are also used, as the Bravais-Pearson or Spearman correlation coefficients, or the Nash-Sutcliffe efficiency (which compares the quadratic error of the model to the variance of the observations). Since 2010, graphical combinations of various metrics (Fig. 6) have been increasingly used: Taylor's diagram showing the Bravais-Pearson correlation coefficient, the RMSE and the ratio between simulated and observed standard deviation (Neumann & Schernewski, 2008; Eilola et al., 2011; Testa et al., 2014; Lessin et al., 2014), target diagram showing the relative bias against with the RMSE (Lopes et al., 2015). When the time course of values is considered as crucial, autocorrelation of the model error time series can be used (Coffaro & Bocci, 1997), because non-persisting bias can be considered as a sign that no crucial process is missing in the model. On the contrary, when the time succession of values does not matter, but only their statistical distribution, (Cerco & Noel, 2013; Picart et al., 2015) have compared the cumulative density functions of the observed and simulated values. The ascertainment arises that there is a general decreasing goodness of fit when one looks at state variables ranging from pure physical ones (temperature, salinity, currents) to biological ones located at a high trophic level. Regarding eutrophication, a clear-cut change arises between inorganic nutrients and the first common biological descriptor, i.e. the total chlorophyll (Allen et al., 2008).

Calibration is followed by model validation, for which (Fitzpatrick, 2009) has defined three types: time validation (the model is run on a period different from the calibration one), space validation (the model is run on a site different from the calibration one) and structure validation (the model must fill some consistency constraints, as the mass conservation, or stay inside some plausible interval). Starting from the example of two models of nitrogen loads into nine North-American estuaries, which gave the same fit to measurements, but estimated the part of urban waste in these loads respectively to 47% or 83%, (Giblin & Vallino, 2003) insist on the space validation on very different sites, as well as on the structure validation by constraining the simulated fluxes by isotope tracking measurements. As pointed out by (Allen et al., 2008), the validation process reveals a common feature of eutrophication models (and more generally ecological models), i.e. their inability to reproduce extreme situations (sudden big blooms or acute hypoxia), which are indeed the most harmful for the ecosystem. A trial of a linear transformation of the simulated values (Picart et al., 2015) does not improve the tails of the distributions of the simulated state variables. In that particular case, authors blamed more the biogeochemical part than the hydraulic one in the weakness of the models, as did (Turner et al., 2014) when they showed the crucial improvement brought by introducing a sedimentary compartment, while (Los & Blaas, 2010) showed in their retrospective of the successive developments of BLOOM/GEM model since the 80s that discrepancies between model results and data were caused primarily by inadequacies in the model forcing: hydrodynamics, suspended sediment concentrations and river loads.

Some studies add to the calibration/validation phase a sensitivity analysis, in order to detect the parameters either over-influent or without any effect on the simulated results. By re-running the model with

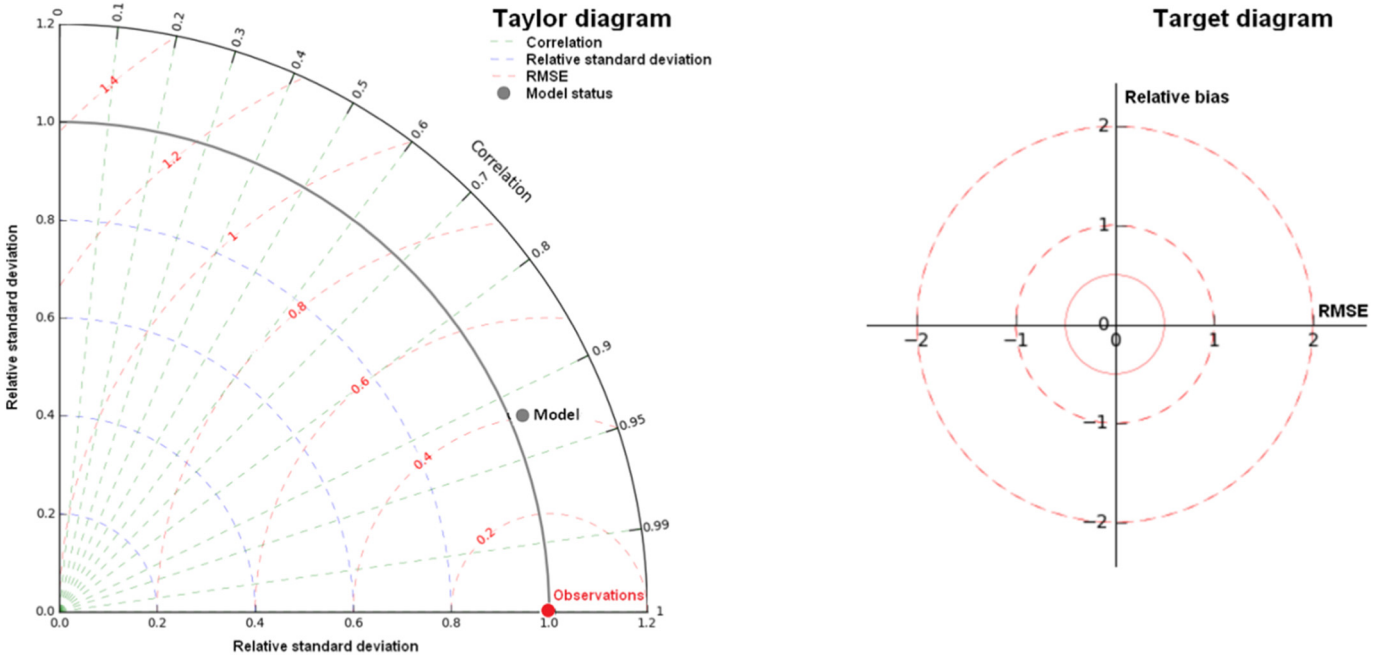

Fig. 6. Taylor and Target diagrams.

different values of each parameter, each parameter of the equations can be characterized by its normalized sensitivity, i.e. the ratio between the relative effect to the relative cause (Chapelle, 1995). The sensitivity to the time resolution of the physical forcing (wind in (Chapelle et al., 2001) or the biogeochemical boundary conditions (marine boundary in (Wild-Allen & Rayner, 2014) is done only when high frequency time-series are available, and allow various sub-sampling. It has revealed the definite incapacity of models to reproduce accurately some transient features in case of a too loose forcing. As soon as the model simulates some long-life compartments (such as detrital matter, or sedimentary pools), a several years long period of spin-up is necessary, either to forget the unrealistic initial condition or to confirm the stability of the realistic one imposed from observations. Up to now, very few models have performed a long spin-up period before the nominal simulation: 5 years for the EU-MT-MIKE21 model used by (Rasmussen et al., 2009), 10 years for the ERSEM model used by (Petihakis et al., 1999), 97 years for the RCO-SCOBI model of the Baltic Sea used by (Eilola et al., 2009).

7. The strength of models: enforcing the land-sea continuum and gaining new information

The first benefit of modelling eutrophication is to disentangle the roles of the main drivers, i.e. the light availability, the confinement and the nutrients. Grazing is generally not considered as a major driver in controlling the course of eutrophication, because the occurrence of high algal biomasses demonstrates the inefficiency of natural grazers; this is especially true in case of mass blooms of free macroalgae, which escape to their usual grazers living on rocky bottoms. But some models have tried to assess the role of grazing at the very beginning of eutrophication process, as well as the efficiency of a man-made harvest of algal biomass (see further discussion about top-down control in the following chapter). Many estuarine models have pointed out the protective role of high turbidity (Cole & Cloern, 1987; Soetaert et al., 1994; Muylaert et al., 2005; Gallegos, 2014; Chen et al., 2014; Mathews et al., 2015). Many others have also explicitly quantified the control of eutrophication by the residence time, either by applying the Vollenweider's formula (Steward & Lowe, 2010), or by comparing various cases (Peterson & Festa, 1984; Liu & de Swart, 2015; Lung, 1988; Wang et al., 2013; Nash et al., 2011; O'Boyle et al., 2015) (Saraiva et al., 2007; Camacho et al., 2015). Some estuarine models (Lin et al., 2008; Buzzelli et al., 2014) show that decreasing the nutrient fluxes by lowering the flow rate may have a worsening effect (increase of residence time and decrease of turbidity, hence increase of phytoplankton biomass. However, (Chan et al., 2002) showed that the historical decrease of flow rate caused by building impoundments along the Swan River had a negligible effect on the onset of eutrophication compared to the concomitant increase of the nutrient concentrations in the river. Modelling in parallel the nitrogen and phosphorus cycles, and taking into account their main sources (rivers, ocean, sedimentary stocks, atmospheric depositions in some cases) has very often confirmed the prominent role of increased nitrogen inputs in triggering and controlling the marine eutrophication process. The case is quite clear in mass proliferations of green algae in Brittany (Ménesguen & Salomon, 1988; Ménesguen, 1992; Perrot et al., 2014; Ménesguen et al., 2006) or in the Venice lagoon (Bendoricchio et al., 1994; Coffaro & Sfriso, 1997; Bergamasco & Zago, 1999; Giusti & Marsili-Libelli, 2005). In the coastal seas, the biggest anoxic sites apart from the Baltic Sea are also nitrogen-controlled: (Evans & Scavia, 2011; Feist et al., 2016; Justic et al., 2002; Obenour et al., 2015; Scavia et al., 2003) show the clear link between increasing N loadings and increasing anoxic area in the Gulf of Mexico. (Cerco & Noel, 2013; Liu et al., 2011; Liu & Scavia, 2010; Stow & Scavia, 2009; Testa et al., 2014) have also shown that in the most saline parts of the Chesapeake Bay, the summer blooms leading to bottom anoxia are controlled by the nitrogen inputs. The degradation of the Chesapeake Bay began two centuries ago, and was enhanced when the top-down control provided by natural huge oyster populations collapsed due to overfishing at the end of the 19th century. Eutrophication models containing filter-feeder state variables provide a unique opportunity to compare the efficiency of bottom-up regulations (e.g. decrease of river nutrient loads) with top-down regulations (e.g. massive re-introduction of oyster farming, where the water quality allows). (Fulford et al., 2007) show that even the huge pristine stock of oysters could not prevent the central channel from excessive primary production and hence, from bottom anoxia, in the present enriched context. For the English Channel and the North Sea also, the models have pointed out the nitrogen triggering of eutrophication phenomena, e.g. the mucus production by the proliferating colonial flagellate *Phaeocystis globosa* (de Vries et al., 1998; Lancelot et al., 2005; Lenhart et al., 2010; Lancelot et al., 2014), but also the current culmination of diatom blooms by the appearing lack of phosphorus or silicon, along with the increase of dinoflagellate blooms (Guillaud & Ménesguen, 1998; Guillaud et al., 2000; Cugier et al., 2005; Passy et al., 2016). The current intensity of summer blooms in the downstream part of many estuaries also appears to be N-controlled (O'Boyle et al., 2015; Gallegos & Jordan, 1997; Timmermann et al., 2010), but some estuaries remain P-controlled, as freshwater systems (Fan & Song, 2014; Liu et al., 2014; Meeuwig et al., 2000; Lung, 1988; Lung & Testerman, 1989). Lagoons also are very sensitive to nitrogen increase (Baird et al., 2003), but the recurrent sedimentation of organic matter in these calm and shallow water bodies (accelerated by the shellfish cultures (Chapelle et al., 2001) creates a large benthic stock of N and P, from which anoxic crises can release sudden bursts of phosphate into the water column (Humborg et al., 2000; Gikas et al., 2009; Schernewski & Wielgat, 2001). The most noticeable exception to the control of marine eutrophication by nitrogen inputs is the Baltic Sea, the northern and eastern parts of it being brackish, with very low salinities in some parts (Bothnian Sea), favorable to cyanobacteria blooms. (Eilola et al., 2009; Savchuk & Wulff, 2007; Savchuk & Wulff, 2009) have simulated the historic spreading of bottom anoxia caused by the increased loads of phosphorus, and deduced from various forecasts that restoration of a good bottom oxygenation could take around a century even in case of drastic nutrient loads reductions.

The power of models lies in their inexpensive capacity to simulate rapidly alternative scenarios to the present one, or not sampled past situations. Most of the models show that a strong abatement of loadings will be necessary to halt eutrophication of sensitive areas, sometimes equal for N and P, as in Chinese coastal waters (Qiao et al., 2014), but very often focused on N loadings now, because the phosphorus ones have been drastically reduced in many developed countries. For the Gulf of Mexico, (Justic et al., 2007) took into account the observed climatic changes and were compelled to revisit their former recommendations (Justic et al., 2003), and to upgrade from 30% to 40–45% the N load reduction necessary to achieve the containment of the hypoxic area under 5000 km^2, as required by the Gulf Hypoxia 2008 Action Plan, designed for reducing, mitigating, and controlling hypoxia in the Northern Gulf of Mexico and improving water quality in the Mississippi River basin. More recently, (Feist et al., 2016; Liu et al., 2010) have pushed this recommendation up to 70%. In the Chesapeake Bay also, (Liu et al., 2011; Liu & Scavia, 2010; Stow & Scavia, 2009) have predicted that the 35% reduction planned for the N loads will not be enough to keep the anoxic volume under 3 km^3. Models offer also an easy way to test simulated land-use scenarios, leading to various forcing by nutrient leakage from continental watersheds. Probably because of the limited area of the watersheds of their tributaries, only two lagoon models have been connected to a watershed model, e.g. SWAT (Plus et al., 2006; Malhadas et al., 2014). Estuarine models are more frequently coupled to a watershed model (Carmichael et al., 2004; Kauppila et al., 2003; Adamack et al., 2012; Meeuwig, 1999; Gypens et al., 2013). Only models of wide coastal zones have increasingly be forced by the outputs of watershed models, used to simulate various scenarios of land use (Lowery, 1998; Cerco & Noel, 2013; Townsend, 2014; Cugier et al., 2005; Lancelot et al., 2005; Lancelot et al., 2009; Lancelot et al.,

2011; Lancelot et al., 2014; Gypens et al., 2009; Passy et al., 2013; Ahlvik et al., 2014; Lancelot et al., 2011; Desmit et al., accepted for publication). As previously shown in the Gulf of Mexico or in the Baltic Sea, (Lancelot et al., 2011; Passy et al., 2016) have shown that in the North Sea, only a scenario of drastic reduction of N river loads, requiring a strong diminution of rearing and fertilization, could restore the coastal water quality in terms of reducing chlorophyll *a* blooms, though with a limited effect on increasing bottom oxygen (Skogen & Mathisen, 2009).

If models have been intensively used to encompass the spatial continuum from watersheds to coastal seas, they have also been used since the beginning (Doi & Nitta, 1991) to explore the trophic continuum, from inorganic nutrients up to top predators. (Barausse et al., 2009; Luang et al., 2014; Nobre et al., 2005) show that moderate eutrophication does not reduce the secondary production, but shortens the pathways and transform a reticulated trophic web into a few short trophic chains, with an increased transfer of organic matter from the pelagic system to the benthos. (Lancelot et al., 2009) have shown also that any decrease in the trophic efficiency (secondary to primary production ratio) related to *Phaeocystis* dominance was the result of an imbalance between DIN and DIP inputs, explained by the reduction in DIP loads after 1985 while DIN remained elevated.

The pervasive constraint to compare their simulated state variables to measured quantities has probably bridled the creativity of modellers. Using validated models to do "better" than, or to go beyond the observations is a recent practice to be encouraged. Models are indeed the only way to accede to unmeasurable descriptors, e.g. the fraction of a chemical element coming from a specific source, and the age of this element, i.e. the mean time elapsed between the release from the source and the arrival in the current marine position. Initially limited to the use of passive tracers put in the rivers and the oceanic boundary (Kasih & Kitada, 2004; Lacroix et al., 2007; Wild-Allen et al., 2010; Eilola et al., 2014; Djambazov & Pericleous, 2015), this approach has recently moved towards the use of active and form-changing variables, e.g. nitrogen passing from dissolved inorganic form to particulate phytoplankton form, then to zooplankton or detrital forms (Ménesguen et al., 2006; Sugimoto et al., 2010; Timmermann et al., 2010; Troost et al., 2013; Los et al., 2014; Dulière et al., 2017; Große et al., 2017). This technique has the advantage of being non-disruptive of the computed biogeochemical fluxes, whereas the straightforward removal of a nutrient source modifies the balance between the fluxes. Another specific capability of models is the automatic computation of an optimal solution of remediation, according to a given target. As optimization techniques are based on a cost function to be minimized, the models can provide a unique opportunity to merge in a unique mathematical cost function biogeochemical and economical objectives, for instance for the Baltic Sea (Laukkanen et al., 2009; Ahlvik et al., 2014) or for the Swedish Limfjord and its mussel farms (Timmermann et al., 2014).

8. The weakness of simulation: what is not yet modelled? challenges and gaps

The early deterministic models, running on slow computers with tiny memory, described the biogeochemical cycles with very few bulk variables: if they were easy to understand and to manipulate, they were considered as having a definitely limited reliability and accuracy. The drastic increase of computer power brought the hope that highly more detailed models (structurally refined and spatially resolved) would automatically improve the reliability of the eutrophication models. The development and the successive improvements of the European ERSEM model is a good example of this trend (Baretta et al., 1994; Baretta-Bekker et al., 1997; Pätsch & Radach, 1997). Surprisingly however, (Cerco & Noel, 2013), comparing with themselves twenty years before (Cerco & Cole, 1993), note that the huge complexification of the model of the Chesapeake Bay has more succeeded in lightening some dark regions of the model by new state variables than in improving substantially the goodness of fit of the basic descriptors! This frustrating apparent limit to the accuracy of deterministic models has regularly pushed some authors to prefer purely empirical models, either statistical (Gowen et al., 1992; Edwards et al., 2003; Tamvakis et al., 2012) or, more recently, based on neural networks (Melesse et al., 2008; Muller & Muller, 2015). The lack of internal processes however limits the application of these models to scenarios or sites not very different from the case study where the calibration has been done, and prevents them to forecast situations where a hidden variable (not explicitly taken as an input of the model) has changed, for instance in the perspective of climate change.

As almost all the models of marine eutrophication start with the postulate that the eutrophication process is bottom-up controlled (i.e. by physics and supply of inorganic nutrients), very scarce are those which explore the complementary role of the top-down control, even if some state variables are taken into account in the model (zooplankton, gelatinous plankton, benthic filter feeders). Apart from (Fennel, 2010; Kolesar, 2006; Lancelot et al., 2002), they focus on the cultivated bivalves only (Fulford et al., 2007; Maar et al., 2010; Grangeré et al., 2010; Timmermann et al., 2014). (Cellina et al., 2003) searched the optimal strategy for the anthropogenic top-down control of green algae mass proliferations through an intensive mechanical collecting; they found that an intensive collecting at the very beginning of the algal growth was the most cost-effective.

Modelling the possible modifications of current eutrophication under climate change remains very uncertain, for instance because the flow rates of big rivers such as the Mississippi can change from −30% to +40% depending on the climatic scenario (Justic et al., 2007). In the Baltic Sea, the unknown long-term behavior of the sedimentary stocks of phosphorus appears as a great factor of uncertainty, even if models agree in prognosing an increase in cyanobacteria blooms (Lessin et al., 2014) and in the size of the anoxic bottom area (Meier et al., 2011; Meier et al., 2012; Eilola et al., 2012; Skogen et al., 2014; Ryabchenko et al., 2016) The effect of sea level rise on marine eutrophication has been rarely studied: (Martins et al., 2008) think that the green algae will follow the changes in the extent of very shallow areas, but (del Barrio et al., 2014) predict a possible regression of submersed phanerogams.

9. Future needs

As eutrophication is a nuisance, its remediation has a heavy cost in financial terms, but also in social consequences. Therefore, the decision-makers would like to know where to focus the nutrient load reductions, and what is the level of confidence in the quantitative recommendations produced by the eutrophication models. The first question needs a more detailed quantitative theory of eutrophication, i.e. mathematical relationships linking all the controlling factors (light, confinement, nutrients) to the various symptoms of eutrophication (high algal biomass, bottom oxygen depletion, toxins, shift in biodiversity). This could improve the risk mapping initiated by the EUTRISK (Druon et al., 2004), and increase its reliability in the frame of climate change. As already done for winter nutrients, inverting these cause-effect relationships could be a powerful way to compute the river-specific reductions of loadings necessary to ensure a desired level of remediation. For the second question, as explained before, an exhaustive assessment of the uncertainty on the results caused by the uncertainty on parameters or forcing variables is beyond the common power of current computers, because of the duration of the thousands or millions of simulations necessary, especially if combined effects of several parameters are studied. Nevertheless, a few models, belonging to the statistical type as well as to the deterministic type, have already weighted their mean recommendations by confidence intervals, even if those have been incompletely computed, and this empirical approach must become more common.

Up to now, the vast majority of the models have been used in a short-term perspective, so they could not tackle long term changes of the marine ecosystems. The future models should take into account

the long term changes in benthic nutriment stocks, as well as the changes in biodiversity possibly induced by an increasing nutrient enrichment and its subsequent decrease. The bulk variables (diatoms, dinoflagellates…) used for algae will probably be replaced by a lot of competing species differing by their ecophysiological parameters, as introduced by (Follows et al., 2007) for the simulation of naturally emerging biogeographical communities, or by size- and trait-based variables used in recent generic models of pelagic ecosystems (Andersen et al., 2015). However, not only bottom-up drivers do control the ecosystem shift during the eutrophication process, some top-down controls may change (e.g. from copepod- to jellyfish dominated foodwebs) and more complex and adaptive descriptions of the trophic web have to be also introduced in eutrophication models.

As previously mentioned, a lot of modelling studies have concluded, from a pure ecological and biogeochemical point of view, that returning to a GES might require drastic changes in the watershed use. As these changes are not easily accepted by the society, it appears, from an operational point of view, that better estimating the feasibility and the cost of various remediation scenarios by coupling socio-economic models to eutrophication models could help in building successful remediation plans.

10. Conclusion

Models of marine eutrophication have appeared about ten years later than the models of lake eutrophication, but their number is still increasing and accompanies the increase of the spots of marine eutrophication, due to the crescent nutrient inputs from rivers, especially nitrogen. This review reveals that deterministic models are strongly dominant over statistical ones, but that they show a very large range in process and space resolution. They have brought many fruitful operational recommendations, by targeting the responsible nutrient sources and assessing the magnitude of necessary load reductions. They must now move from short-term simulations towards secular ones, and better take into account the environmental accumulators of nutrients (sediments in marine models, underground water and soils in continental models). Introducing numerous types of algae and trait-based foodwebs in these long-term models will not only add complexity to models, but probably allow some qualitative jump in the simulation of dynamic properties of the ecosystems. This will bring new perspectives in forecasting the changing biodiversity resulting from the onset of eutrophication, as well as its regression. Coupling with climate change scenarios will then be more fruitful. Models have had always a heavy tribute to field observations: they will obviously take advantage for their forcing and their calibration/validation from the increase of high frequency automated sensors. At the end of this review, it appears that designing new sensors for measuring the "hidden" state variables of the models (e.g. detrital forms in water and sediment) would certainly create a qualitative improvement in the calibration of models, and hence their reliability for operational recommendations.

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